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Original article

Spatial prioritization for urban Biodiversity Quality using biotope maps and expert opinion

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ABSTRACT

Spatial prioritization can produce useful information about biodiversity values from urban areas. However, its typical focus on (endangered) species distributions assumes a rather restricted approach to urban biodiversity. In 2006, Feest suggested that five attributes of species assemblages more holistically describe the so called "Biodiversity Quality" of an area: species richness, biomass, population density, evenness, and rarity. Here we apply these attributes in spatial prioritization for urban biodiversity, across ten taxonomic groups: vascular plants, polypores, fungi (other than polypores), birds, bats, mammals (other than bats), herpetofauna, butterflies, hymenoptera, and beetles. In addition, we introduce two more attributes relevant for urban biodiversity conservation: support for specialist species and regional representativeness of the species assemblages. First, spatial data about local urban biotopes was acquired. For each taxon, the capacity of each urban biotope to support the seven introduced attributes of Biodiversity Quality was evaluated via expert elicitation. Expert opinion was then translated into a spatial analysis implemented with the Zonation software. Different anthropogenic, semi-natural, and natural habitats, such as herb-rich forests, lakeshores, open wastelands, fortifications, and botanical gardens, were identified as important for urban Biodiversity Quality. To minimize negative impact on biodiversity, future construction and development should be directed to built-up areas and agricultural fields. Our conception of urban biodiversity lies in between species- and habitat/ecosystem -based analyses and offers a more comprehensive perception of urban biodiversity than a focus on species distributions only, which facilitates the planning of ecologically sustainable cities and biodiverse urban green infrastructure.

1. Introduction

Ensuring that cities grow in a biodiversity-friendly way is a major conservation issue in the urbanizing world (Marzluff, 2002; Miller and Hobbs, 2002; Ricketts and Imhoff, 2003; Dearborn and Kark, 2010; Seto et al., 2012; McDonnell and Hahs, 2013; Shwartz et al., 2014; Soanes et al., 2018). The conservation of urban biodiversity is, however, a challenging task compared to conservation in rural areas (Battisti and Gippoliti, 2004; Kowarik, 2011; McDonnell and Hahs, 2013; Shwartz et al., 2014; Soanes et al., 2018). In cities, ubiquitous human activities and impacts cannot really be separated from urban biodiversity, which voids the typical conservation aim of avoiding or removing human impacts (Dearborn and Kark, 2010; Kowarik, 2011; Rupprecht et al., 2015; Pickett et al., 2016; Soanes et al., 2018). Instead, planners often focus on the maintenance of diverse and sustainable urban ecosystems that bring multiple benefits to urban residents and constitute an integral part of urban green infrastructure (Battisti and Gippoliti, 2004; Tzoulas et al., 2007; Goddard et al., 2010; Ahern, 2013; McDonnell and Hahs, 2013; Hansen and Pauleit, 2014; Shwartz et al., 2014; Parker, 2015; Nilon et al., 2017; Capotorti et al., 2019; Jerome et al., 2019). The question of how urban biodiversity is interpreted and measured by practitioners in the realms of planning, management, and conservation is important (Ahern, 2013). As described by Pickett et al. (2016), urban biodiversity can be understood as biophysical patterns that occur in cities (biodiversity-in-the-city paradigm), as an inseparatable combination of human and non-human parts of urban ecosystems (biodiversity-of-the-city), or as a key component in achieving urban sustainability with multidiciplinary cooperation (biodiversity-for-the-city). Regardless of the chosen paradigm of urban biodiversity, it is important that the underlying measure of urban biodiversity is appropriate for the intended applications of the analyses.

Urban biodiversity analyses are often based on habitat mapping that

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can be generalized over the entire city area using e.g. satellite images (Hand et al., 2016; Farinha-Marques et al., 2017; Nilon et al., 2017; Li et al., 2019). Habitat maps can be complemented with data about species or other taxonomic units (Hand et al., 2016; Li et al., 2019) or e.g. residents' appreciation of green areas (Wang et al., 2019). In Europe, the term biotope is often used to refer to distinctive urban habitat types that support similar species assemblages (Sukopp and Weiler, 1988; Müller, 1998; Müller and Fujiwara, 1998; Löfvenhaft et al., 2002; Qiu et al., 2010). Löfvenhaft et al. (2002) provide a nice overview of the biotope concept, which is very similar to the concept of habitat types. Biotope mapping can be based on e.g. aerial imagery (Löfvenhaft et al., 2002) or field inventories (Sukopp and Weiler, 1988). Biotope or habitat mapping is a common tradition in Europe and New Zealand in particular (Freeman and Buck, 2003; Qiu et al., 2010).

The conservation of urban biodiversity requires effective planning, due to many conflicting land-use interests and high land price (Haaland and van den Bosch, 2015). Spatial (conservation) prioritization refers to a set of methods for identifying priority areas for biodiversity conservation in an efficient, cost-effective, and transparent way (Bekessy et al., 2012; Lehtomäki and Moilanen, 2013). Spatial prioritization should be applicable to urban biodiversity conservation planning (Gordon et al., 2009; Bekessy et al., 2012). In order to generate policy-relevant solutions, the objectives of the prioritization should be clearly stated and justified a priori, and supported both by data and the implementation of the prioritization analyses (Ferrier and Wintle, 2009). This is especially important in growing cities, where inappropriately chosen conservation objectives for biodiversity might result in the rapid loss of areas that would be highly prioritized according to other, better-justified, objectives.

Most spatial prioritizations utilize species or habitat data (Kullberg and Moilanen, 2014). Such analyses effectively aim to maintain habitats, species and populations, either implicitly or explicitly, giving higher priority to locations that support many rare species or habitats (Kujala et al., 2018a). Likewise, urban prioritization studies have typically been performed using species data, or data about both species and habitats (Gordon et al., 2009; Rebelo et al., 2011; Bekessy et al., 2012; Whitehead et al., 2014; Karimi et al., 2015, 2017; Onikura et al., 2016; Albert et al., 2017; Cimon-Morin and Poulin, 2018; Lu and Qu, 2018 – but see Crossman et al., 2007). Species preservation has been the stated goal of some urban prioritizations (Gordon et al., 2009; Onikura et al., 2016); others have used species data as a surrogate for broad ecological values in socio-ecological prioritization (Whitehead et al., 2014; Karimi et al., 2015).

It can be questioned whether a species-centric approach to spatial prioritization provides adequate information about biodiversity in the city, if the general objective of the analyses is to locate diverse ecosystems and associated ecosystem processes to support conservation and sustainable planning in growing cities (Andersson, 2006; Ahern, 2013; Jerome et al., 2019). Basing prioritization on rare and endangered species may lead to the neglect of common biodiversity (Gaston, 2010), which fulfils many ecosystem processes (Gaston and Fuller, 2008; Winfree et al., 2015). If the prioritization is based on species of national interest, the resulting priority areas may favor natural remnants at the expense of novel human-modified urban habitats, which contribute greatly to urban biodiversity (Knapp et al., 2008; Kowarik, 2011; Rupprecht et al., 2015; Soanes et al., 2018). Furthermore, due to the high levels of anthropogenic pressures, urban areas may well suffer from an extinction debt, especially regarding rare species (Hahs et al., 2009). We therefore propose that spatial prioritization for urban biodiversity would generally benefit from a broader approach than just concentrating on endangered species conservation.

Feest (2006) and Feest et al. (2010) proposed an approach called "Biodiversity Quality" for measuring and monitoring biodiversity more comprehensively than mere species richness. Biodiversity Quality is a combination of community attributes that relate to both sustainable ecological communities and resilient ecosystem processes (Hooper et al., 2005; Feld et al., 2009; Feest et al., 2010; Biggs et al., 2012; Harrison et al., 2014), which are at the core of urban biodiversity conservation and sustainability discourses (Andersson, 2006; Ahern, 2013; Parker, 2015). The approach uses a set of five species assemblage metrics: richness, biomass, density/population size, evenness/dominance, and rarity or some other index of species value (Feest, 2006). Together, these metrics describe the characteristics of local biodiversity found in a given site (Feest et al., 2010). These metrics should be reported and used jointly, because they provide complementary information about species assemblages (Feest et al., 2010). The Biodiversity Quality approach is practical, and its applicability has been demonstrated with different taxa, including macrofungi (Feest, 2006, 2009; Feest et al., 2010; Ambrosio et al., 2018), bryophytes (Feest et al., 2010), butterflies (Feest, 2006; Feest et al., 2010, 2011, 2014), carabid beetles (Feest et al., 2012), spiders (Feest and Cardoso, 2012), and birds (Murata and Feest, 2015). However, comprehensive and systematic monitoring of urban species assemblages is very laborious and rarely assessed at the whole city scale (Nilon et al., 2017). Therefore, consultations with experts regarding their opinions on Biodiversity Quality is a relevant option (Jalkanen and Vierikko, 2018). Such approaches have often been used in conservation planning (Martin et al., 2012).

Here, we introduce a method for spatial prioritization that builds upon the framework of Biodiversity Quality. Instead of analyzing the species assemblages of individual areas, we first gather pre-existing data about the distributions of local urban biotopes. Then, we use expert elicitation to estimate the potential value for Biodiversity Quality provided of each urban biotopes for different taxa. Spatial prioritization allows the identification of priority sites for the biophysical realm of urban biodiversity. Our method can also be expanded to cover the human dimension of urban areas, thus enabling more holistic prioritizations of urban biodiversity (Pickett et al., 2016).

2. Materials and methods

2.1. Study area

Our study was performed in the Helsinki Metropolitan Area, located on the southern coast of Finland. The area consists of four municipal cities, Helsinki, Espoo, Vantaa, and Kauniainen, and covers roughly 770 km^2. With its 1.2 million inhabitants, it is the most populous urban area in Finland (Statistics Finland, 2019). In the European context, the urban structure of the Helsinki Metropolitan Area is sprawling with a smallish urban core (Kasanko et al., 2006). There remains many green areas, some even in semi-natural condition, in the area. The area is rich in different types of urban and natural forests, aquatic environments, and agricultural areas (Vierikko et al., 2014, see also Fig. 2). The area has a large rural-like urban fringe, with extensive forests and agricultural fields, but also some relatively extensive semi-continuous forest areas that extend almost to the urban center.

The Helsinki Metropolitan Area is currently one of the fastest growing urban areas in Europe and its population is expected to grow by over 600,000 (~ 40 %) by 2050 (The Helsinki City Plan 2050, 2016). Despite strong pressures for growth, the value of green areas and urban biodiversity is also recognized by the planning authorities (e.g. The Helsinki City Plan 2050, 2016), which emphasizes the need for analyses such as the present one.

2.2. Overview of the approach

Our main challenge in the spatial prioritization of urban Biodiversity Quality was the acquisition of spatial data on different community attributes from our study area. To acquire such data, we used pre-existing biotope-mapping and expert opinions. Fig. 1 summarizes the structure of the approach, which has eight phases: (1) development of the urban biotope classification (Section 2.3); (2) derivation of an urban biotope map that corresponds with the biotope

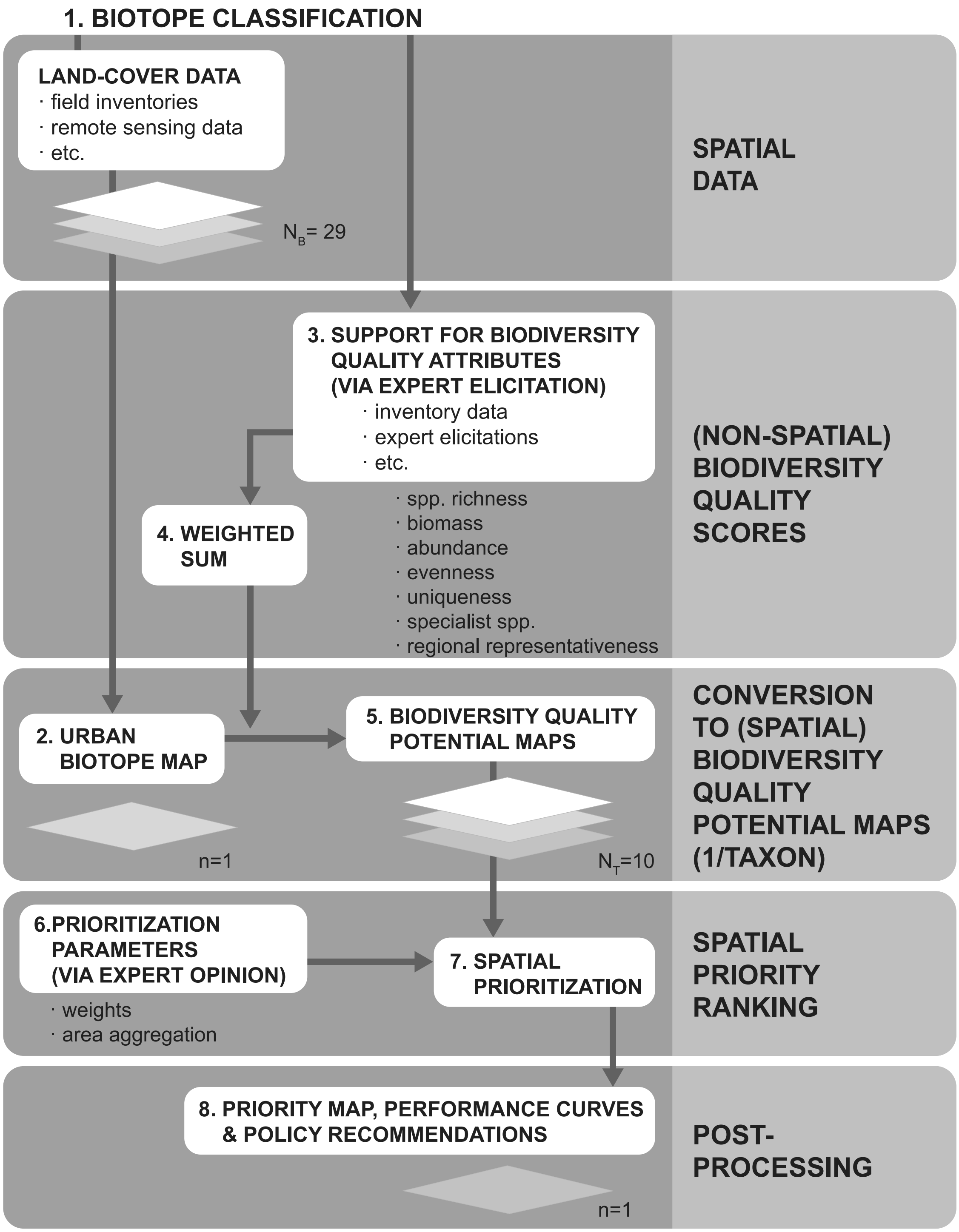

Fig. 1. Workflow for the proposed approach to conservation prioritization in urban environments. N_B refers to the number of available data sources on urban biotope distributions, and N_T to the number of taxa included in the analysis.

classification (Section 2.3); (3) collection of data on the extent of support of each biotope for different Biodiversity Quality attributes of different taxa (Section 2.4); (4) calculation of the weighted sums of the attribute values into Biodiversity Quality scores (Section 2.4); (5) conversion of the biotope map into Biodiversity Quality potential maps for individual taxa (Section 2.4); (6) definition of weighting and connectivity parameters (Section 2.5); (7) the computational spatial prioritization itself (Section 2.5), and (8) interpretation and validation of results. In phase 2, biotopes were evaluated separately for multiple Biodiversity Quality attributes, such as species richness and biomass.

2.3. *Urban biotopes as the basis of the analysis (phases 1–2)*

The spatial mapping of biotopes (Fig. 1, phase 2) is a compulsory pre-requisite for our approach, which we propose as a robust and feasible approach to the prioritization of urban Biodiversity Quality. We based the analysis on biotopes rather than inventories or modelled data for species or communities, because biotopes can be easily defined with limited effort and their number is considerably smaller than the number of species, as a result of which they are generally easier to map than species distributions. Finland has a long tradition in biotope mapping,

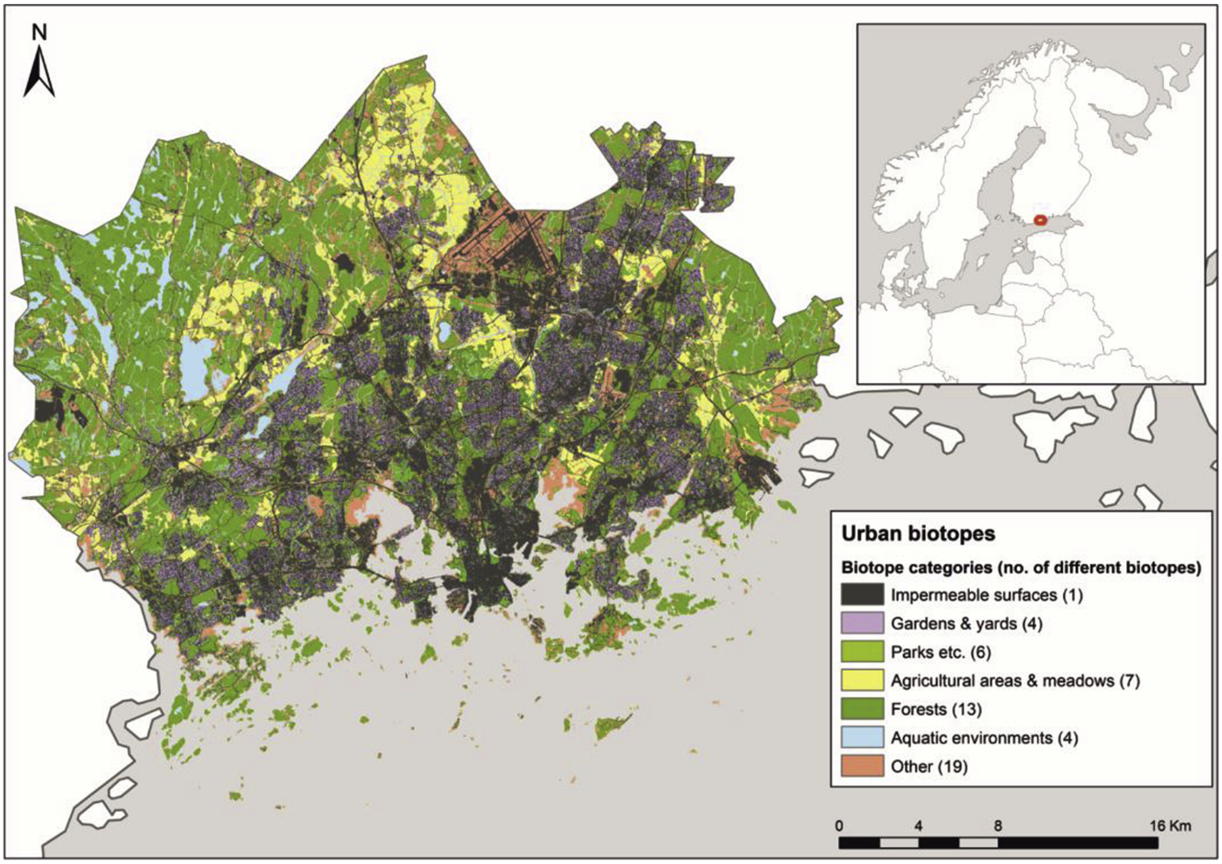

Fig. 2. A simplified urban biotope map of the Helsinki Metropolitan Area. In this map, the biotopes are grouped in 8 distinctive biotope categories of roughly the same size for the sake of visual clarity. Most of the individual biotopes have comparatively narrow distributions and they would have been poorly visible in a single map. See Appendix 1 for the complete list of urban biotopes.

and detailed urban biotope classifications (Fig. 1, phase 1) are available for the Helsinki Metropolitan Area (e.g. Vierikko et al., 2014; Jalkanen and Vierikko, 2018). Furthermore, because biotopes usually also involve the abiotic environment and management characteristics, we considered them as surrogates for different urban ecological communities. The price to pay for the simplicity and good measurability of biotopes is that working with biotopes only offered a general view of the *potential* for Biodiversity Quality.

We used an urban biotope map (Fig. 1, phase 2) of Jalkanen (2016). The map covered all relevant urban biotopes, anthropogenic and natural habitats (varying from e.g. private gardens to brownfields and urban forests) and both private and public lands. The resulting biotope map included 54 different urban biotopes (Appendix 1; Supplementary information S1), mapped at a 20 m resolution. Fig. 2 shows a simplified representation of the map.

2.4. Expert elicitation about support for Biodiversity Quality attributes (phases 3–5)

Next, we carried out the expert elicitation (Fig. 1, phase 3) to examine how the urban biotopes mapped in the previous phase support different attributes of Biodiversity Quality (Feest et al., 2010) for the selected taxa. The evaluated attributes were:

i level of *species richness* of the species assemblage typically supported by the biotope,
ii total *biomass* of the taxon supported by the biotope,
iii *abundance* of the taxon usually supported by the biotope,
iv *evenness* of occurrence between different species within the taxon, and
v *uniqueness* of the species assemblage typically supported by the biotope. This attribute refers to the *rarity* index (Feest, 2006; Feest et al., 2010) generalized for biotopes. The level of uniqueness describes whether the biotope harbors species assemblages that are typically not found in other urban biotopes.

In addition, we included two more attributes that we propose to be relevant in urban biodiversity conservation:

i support for *habitat specialist species*. Because cities include many ecological filters favoring generalists (Aronson et al., 2016), high support of habitat specialists indicates well-preserved and possibly rare biotopes, and
ii the level at which the biotope's species assemblages can be considered as *regionally representative*. Cities include many biotopes that are also found in nearby rural areas, possibly under different management and certainly in an ecologically different spatial context. Representativeness describes how meaningful the urban areas are for the preservation of those species assemblages within the regional context. For example, urban forests in Helsinki are considered to host much more deadwood and mature trees than nearby rural areas, where forests are managed production forests (Vierikko et al. 2014). Consequently, the urban forests of Helsinki show a high level of naturalness of polypore assemblages (i.e., high polypore representativeness).

A total of 24 local taxonomic experts participated in our elicitation. Experts were chosen so that each taxon had more than one elicitor, and each expert had acknowledged experience on the local species assemblages. Each expert evaluated independently in a web poll how well biotopes support the Biodiversity Quality attributes of their own taxonomic group: vascular plants, polypores, fungi (other than polypores), birds, bats, mammals (other than bats), herpetofauna, butterflies, hymenoptera, or beetles. The number of experts per taxon varied between 2–3, and the scoring was on a scale of 0–4 (4 being the highest). In addition, experts were asked to rate the general confidence to their scorings, separately for each biodiversity attributes. Confidence could be rated as "very unconfident", “unconfident”, “somewhat unconfident”, “somewhat confident”, and “very confident”. See Jalkanen and Vierikko (2018) for the elicitation details, including definitions and precise questions given to taxonomic experts.

We calculated the scores for single Biodiversity Quality attributes (richness, biomass, etc.) as an average of the scores given by experts (ranging from 0 to 4), weighted by the self-rated confidence of scores, separately for each taxon. To emphasize confident answers over nonconfident ones, the confidence coefficient could be 0, 1, 2, 4, or 8, corresponding to "very unconfident", “unconfident”, “somewhat unconfident”, “somewhat confident”, and “very confident” answers, respectively. Thus, the final scores for individual attributes could range from 0.0 (extremely low potential and/or very unconfident answers) to 32.0 (high potential and very confident answers).

Biotope quality potential scores were then calculated as the weighted sum of the attribute scores (Fig. 1, phase 4). The weights were set based on the relevance of the attributes for the maintenance of populations and ecological functioning generated by the taxon (Table 1). They were defined by the same taxon-specific experts that scored the biotopes in an expert workshop. Lastly, we created “Biodiversity Quality potential maps” (Fig. 1, phase 5), one for each taxon, by reclassifying the categorical biotope value in the urban biotope map to the respective Biodiversity Quality potential score.

2.5. Spatial Prioritization using the Zonation software (phases 6–7)

We implemented the spatial prioritization using the Zonation software (Moilanen et al., 2011; Lehtomäki and Moilanen, 2013), using the Biodiversity Quality potential maps as input features (Fig. 1, phase 7). Zonation iteratively produces a complementarity-driven ranking of the landscape and tries to maintain a balanced coverage of all input biodiversity components throughout the ranking, thereby ensuring balancing (complementarity) between different taxa.

Zonation employs several alternatives for defining how the balance between features is implemented during priority ranking. We used the *Additive Benefit Function* (ABF) method of ranking because it somewhat emphasizes feature richness compared to other ranking options (Lehtomäki and Moilanen, 2013). This method is appropriate for our goal of identifying areas that are equally important for many communities. Furthermore, ABF is the most appropriate when input features act as surrogates for high overall biodiversity – in our case, all species and ecological communities (Lehtomäki and Moilanen, 2013).

To account for inter-feature connectivity (Fig. 1, phase 6), we used the matrix connectivity method in Zonation (Lehtomäki et al., 2009; Lehtomäki and Moilanen, 2013), which emphasizes aggregations of high-quality areas within a taxon-specific distance at which populations start declining due to fragmentation (Table 2). (The distance values were also defined in the expert workshop). Furthermore, to balance between local habitat quality and connectivity, we followed the procedure of Lehtomäki and Moilanen (2013) and conducted the prioritization using two sets of the ten Biodiversity Quality potential maps. The first set involved the original Biodiversity Quality potential maps representing local habitat quality, and the second set was transformed for connectivity. The weighting between taxa (Fig. 1, phase 6) was set collaboratively by all the experts who participated in the workshop and it was based on the importance of the taxonomic group for the local urban ecosystem (Table 2).

Table 2
Spatial aggregation values (i.e. distance from which populations start suffering from fragmentation) and weights used in the Zonation prioritization. Values were defined in an expert workshop.

Taxon	Distance for spatial aggregation (m)	Weight (total: 100)
Vascular plants	20	40
Polypores	80	9
Fungi	40	13
Birds	100	4
Bats	200	1
Mammals	100	2
Herpetofauna	200	1
Butterflies	50	9
Hymenoptera	50	12
Beetles	40	9

All Zonation settings used in this study, and the associated files, can be found in Zenodo data repository (Jalkanen, 2020).

3. Results

The expert scores for the suitability of different urban biotopes for the Biodiversity Quality attributes of different taxa can be found from the Zenodo data repository (Jalkanen and Vierikko, 2018). The scores for biodiversity attributes of taxa, calculated as weighted averages of expert scores, ranged from 0.0 (impermeable surfaces were set as 0 for every feature, see Jalkanen and Vierikko, 2018) to 24.0 (reed beds and coastal meadows, both for specialist birds). Averaged across all the expert answers, the highest values were given to old-growth herb-rich forests (10.02), herb-rich forests (9.17), fortifications (8.26), old-growth heathland forests (8.08), and botanical gardens (7.77), whereas apartment suburb yards, townhouse gardens, apartment block yards, artificial shores, and impermeable surfaces were given the lowest values (2.89, 2.86, 2.59, 1.91, and 0.0, respectively). Based on confidence ratings given by the experts themselves, biomass was the attribute for which confidence was lowest (average confidence was 2.25 on a linear scale of 0–4; 0 referring to “very unconfident”, 1 referring to “unconfident”, etc.), whereas species richness and specialist species were attributes for which confidence was found to be highest (3.0 for both). (Note that lack of “confident” category may have homogenized the confidence rating averages. Some answers might have been “confident”, were it given as an option, but were instead selected to be “somewhat confident”). The scoring of biotopes was then converted to attribute maps and summed into Biodiversity Quality potential maps (Section

Table 1
Weights given to the single attributes. The sum of attribute weights in each column is 100.

	Vascular plants	Polypores	Fungi	Birds	Bats	Mammals	Herpetofauna	Butterflies	Hymenoptera	Beetles
Richness	20	20	25	27	20	17.5	20	25	25	30
Specialists	15	25	25	18	20	20	15	30	23	20
Biomass	20	10	5	6	15	10	15	3	10	5
Abundance	15	7.5	15	28	20	20	20	3	15	5
Evenness	10	7.5	14	8	15	10	5	2	7	10
Uniqueness	15	10	9	7	5	17.5	5	25	13	15
Representativeness	5	20	7	6	5	15	20	12	7	15

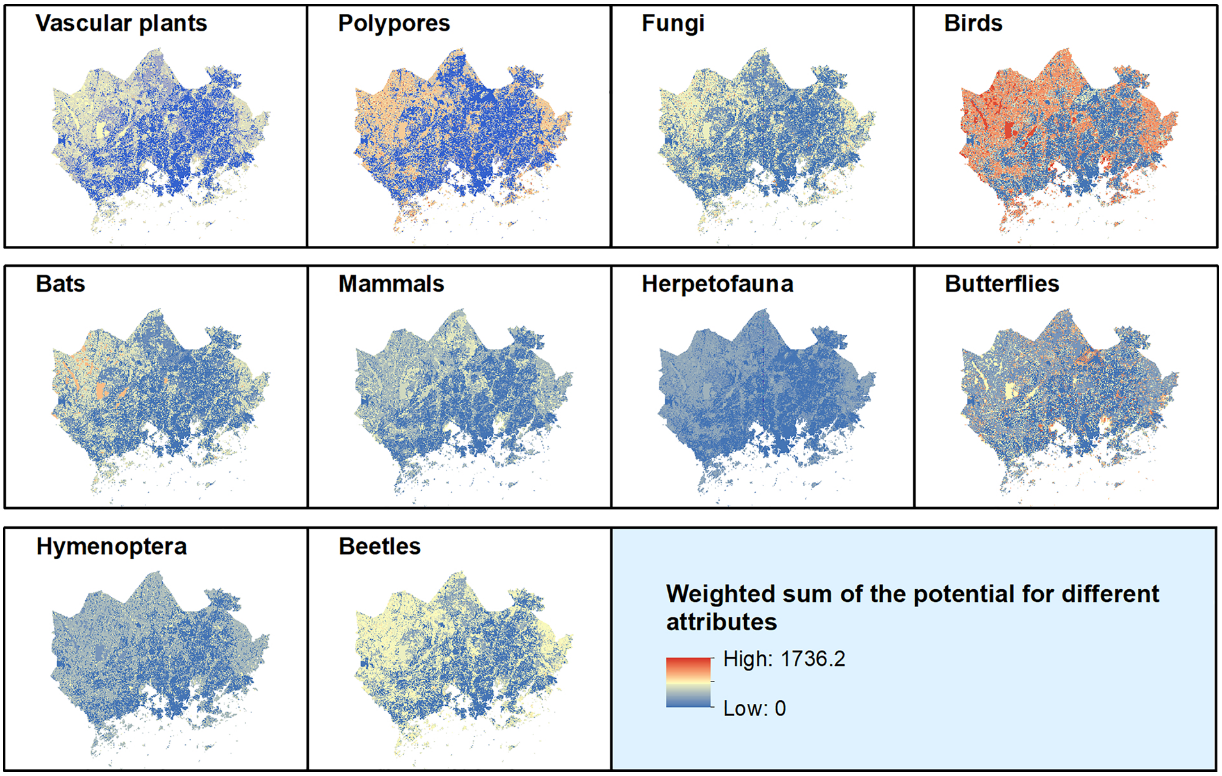

Fig. 3. Biodiversity Quality potential maps that were used as input layers in the Zonation analysis. Birds and polypores show the highest values, whereas herpetofauna show the narrowest distributions and lowest maximum values – suggesting that the Helsinki Metropolitan Area holds the lowest potential for herpetofauna communities.

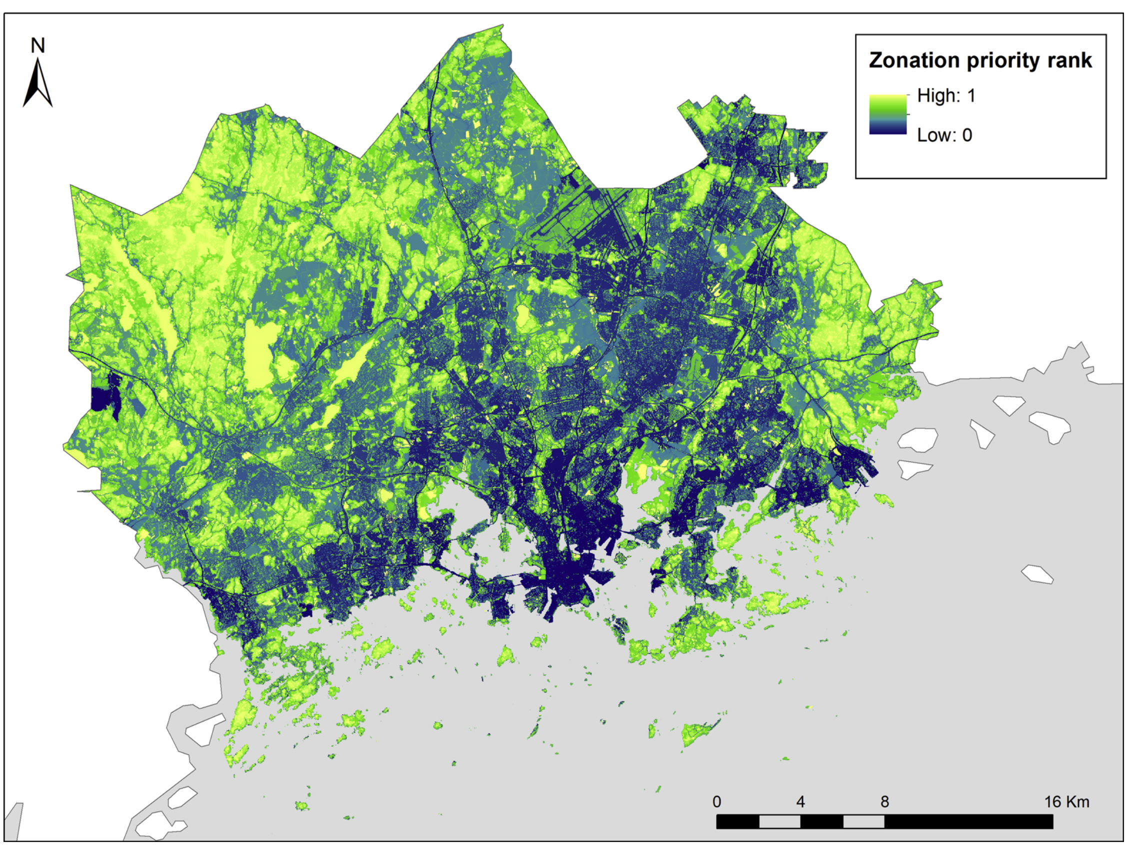

Fig. 4. Zonation output, the priority rank map. The values of the raster image range between 0–1, describing Biodiversity Quality priority rank (with 1 being the highest).

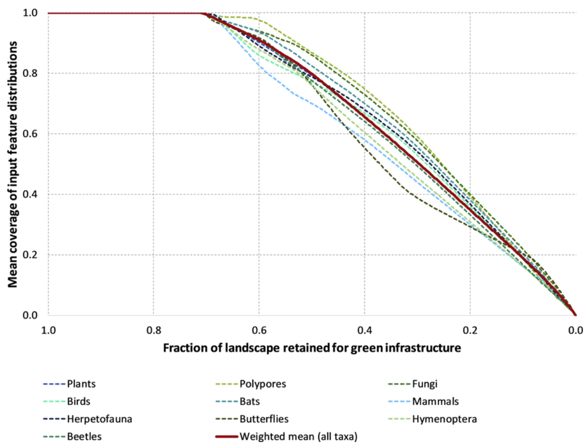

Fig. 5. Zonation representation curves for different taxonomic groups. The drop lines show that the top 20 % of the Helsinki City area would cover almost 35 % of known ecological values that support urban biodiversity.

2.4). In general, the "distribution" sizes were of comparable size for most attributes and taxa, except for the herpetofauna, which are less supported by the urban landscape (Fig. 3).

Fig. 4 shows the Zonation priority rank map, based on the prioritization of the 20 Biodiversity Quality potential maps with matrix connectivity included. The ecologically high-priority areas are favorable for new development in terms of impact avoidance, and vice versa. The largest high priority areas are found in the large forest areas of the northwestern and northeastern parts of the urban fringe. Nevertheless, high value areas are also located in inner-city areas. The top-priority areas include many occurrences for biotopes that received high scores from the experts, such as herb-rich forests, old fortifications, botanical gardens, and lakes. However, due to complementarity and connectivity, they also include e.g. large coastal islands, open ruderal areas, coastal meadows, allotment gardens, and areas beneath power-lines. On the other hand, areas of lowest priority include mainly impermeable surfaces, residential areas, and agricultural fields. All Zonation output files can be found in Zenodo data repository (Jalkanen, 2020).

Zonation also generates representation or performance curves, which show the proportion (of occurrences) remaining for input features through the priority ranking (Lehtomäki and Moilanen, 2013). Fig. 5 shows the performance curves of different taxa. All taxa retain their full distribution until roughly 30 % of the landscape has been ranked (excluded), which corresponds to the amount of impermeable surface in the Helsinki Metropolitan Area. After 30 %, occurrence levels of taxa decrease to an approximately linear relationship with area, although a slight steepening of the curves can be seen after approximately 40 % of ranking. In general, this shape of performance curves shows that Biodiversity Quality is supported comparatively evenly across the Helsinki Metropolitan area. (If biodiversity was highly aggregated, the curves would display a much more strongly convex shape.)

Our study is of utility for both land-use and green infrastructure planning in the Helsinki Metropolitan Area. Ecosystem-wise, new construction should focus on sites of lowest priorities (Gordon et al., 2009; Kareksela et al., 2013). Focusing on areas of lowest 40 % priority (Fig. 6), which include infill-development and some agricultural fields near urban areas, would cause on average only 9 % loss of biodiversity-supporting areas. Acknowledging that some of these areas might have ecological values that are underrepresented in our results (e.g. some of

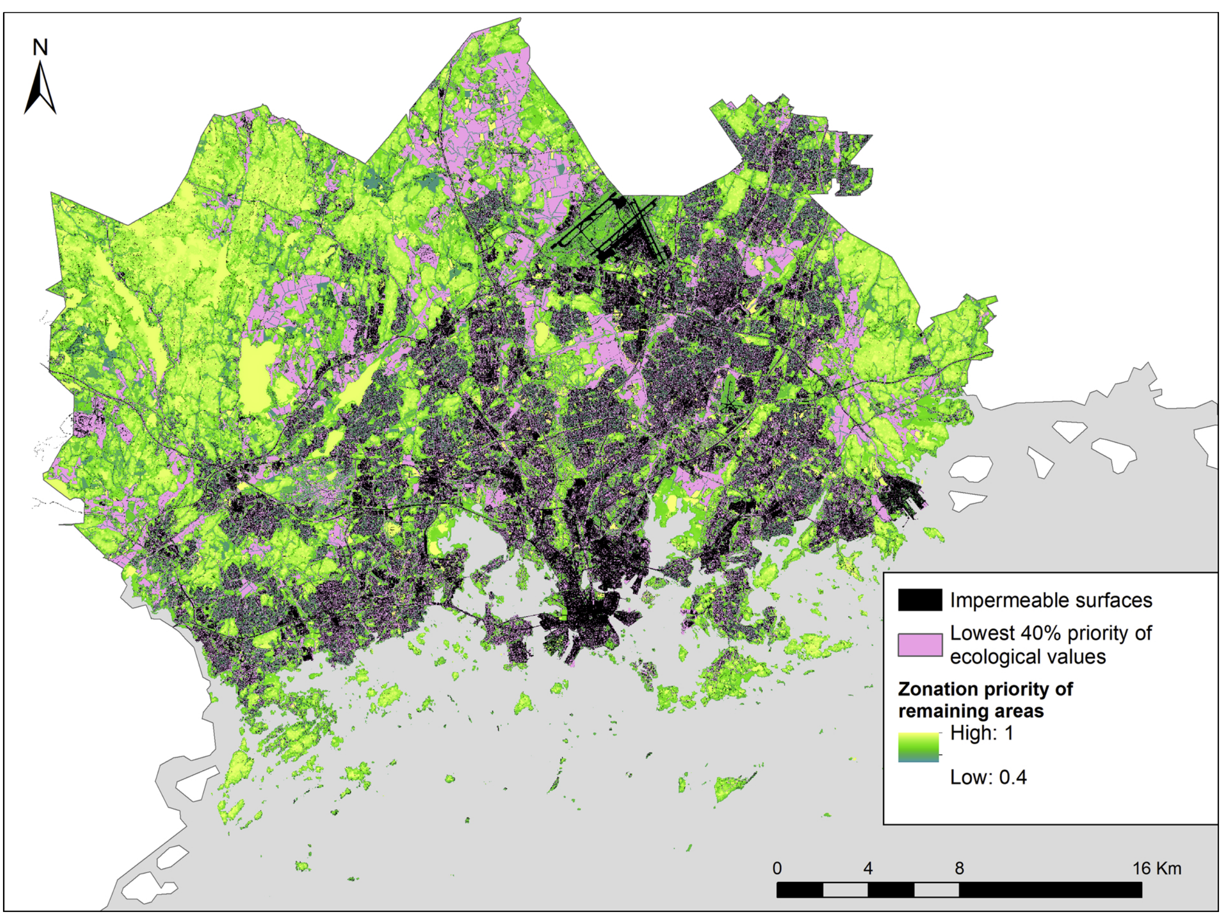

Fig. 6. The lowest-priority areas (in pink) are in terms of impact avoidance most suitable for new development.

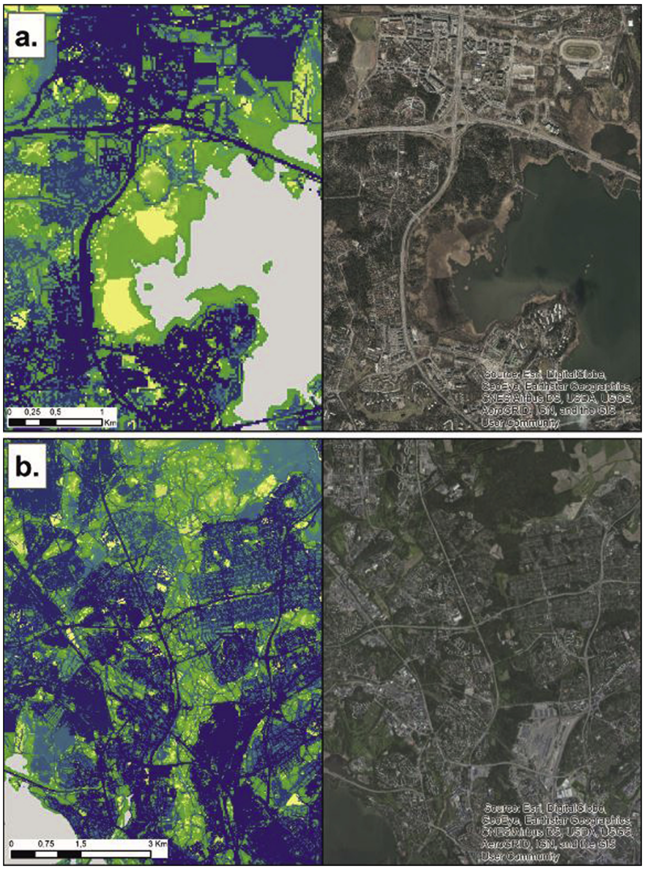

Fig. 7. High-priority areas in (**a**) the Laajalahti bay and (**b**) the Central Park of Helsinki can be clearly distinguished from the surrounding urban areas and would very likely provide high potential for 'socio-ecological hotspots', if planned and managed properly.

the local fields near the coast are considered highly important for bird migration) or other land-use limitations, such as social and cultural values, they can nonetheless be considered the 'safest' focal areas biodiversity-wise to start the planning of new construction in the Helsinki Metropolitan Area.

These analyses can also be used to identify some socio-ecologically important areas for green infrastructure planning. For example, the Laajalahti bay (Fig. 7a) is a large high-priority area with some old-growth forests and coastal meadows in the middle of dense (and growing) urban areas. There is high potential for a socio-ecological hotspot of high value for both biodiversity and visitor appreciation. At the same time, there may be high risk of ecological degradation due to heavy recreational use. Another example can be found in the Central Park of Helsinki (Fig. 7b), which can clearly be distinguished from its surroundings as an elongated area of uniformly high biodiversity priorities. Planning should preserve and enhance the ecological coherence of this area. More detailed examination about the socio-ecological values of different sites than these rather tentative ones would naturally require the inclusion of social data in the Zonation analysis.

4. Discussion

Here we demonstrate a protocol for spatial prioritization of urban biodiversity that is based on the framework of Biodiversity Quality (Feest, 2006; Feest et al., 2010). Rather than using distribution maps of individual target species or habitats as inputs for the analysis, this approach builds on distributions of different attributes of Biodiversity Quality, such as richness and abundance of the urban species assemblages. Our approach provides a representation of biodiversity values throughout an entire metropolitan region, thus supporting sustainable urban planning as well as the planning and management of a biodiverse urban green infrastructure (Jerome et al., 2019).

Acquiring reliable information on species assemblages and Biodiversity Quality attributes is a laborious task (Feest, 2006; Lehtomäki and Moilanen, 2013), which makes it the main limitation of the present approach. Here we used pre-existing spatial data on biotopes and expert elicitation to estimate how biotopes support the Biodiversity Quality of different taxa. Expert elicitation is a widely-used method for obtaining information when data is otherwise sparse (Martin et al., 2012). The main limitation of expert elicitation is the inevitable subjectivity that arises when experts are asked rather general-level questions. Many attributes, such as biomass and uniqueness, can be difficult to evaluate and are prone to differences in individual interpretations. Nevertheless, in many places, expert elicitation may be the only available source of information, especially for less-studied taxa such as fungi or insects. Furthermore, the utilization of expert opinion in the approach presented here is made easier by the fact that, because spatial prioritization is about ranking and comparing, eliciting absolute numbers on Biodiversity Quality metrics is not necessary; it is sufficient to use relative scores. Asking the experts to indicate their confidence in their estimates is important for allocating resources for both conservation (preserve those areas that are certainly important for many taxa), and future biodiversity inventories (focus on taxa or attributes for which good data is missing). In our case study, the confidence options given to experts were biased towards the "unconfident" end of the scale because "confident" was not included as an option for the experts. This bias should not affect the priority ranking (Fig. 4), because Zonation always includes the normalization of the input layers and the bias was systematic over all experts. However, should the approach be applied elsewhere, it would be preferable to use an unbiased scale for the evaluation of confidence.

Our expert scoring was based on pre-existing biotope maps that were available for our study area, albeit scattered across many administrative sources (Jalkanen, 2016; Supplementary information S1). However, prioritizations of urban Biodiversity Quality do not have to rely on biotope mapping. Other types of habitat maps can work equally well, if they comprehensively cover the entire focal area, and the habitat classification is ecologically relevant for the target taxon. These sources can include habitat maps based on vegetation characteristics (Farinha-Marques et al., 2017), automatically-generated habitat maps based on environmental attributes (Li et al., 2019), or, in some cases, even land-cover maps with coarse classification, such as Corine Land Cover (Büttner et al., 2004). Assessment of Biodiversity Quality attributes does not have to be based on the same habitat classification for all taxa; here we used the same classification for the sake of simplicity. If a comprehensive mapping of biotopes would be desired, it should cover the entire city area, it should be based on various biophysical features of urban landscape (e.g. temperature, soil properties), as well as management history of different sites, and it should be done for land parcels no larger than a few hectares (Sukopp and Weiler, 1988). We emphasize that if prioritizations of urban biodiversity are based on general-level biotope or habitat maps without information of the local habitat quality, and especially if they are coupled with expert opinion, the approach is most suitable for summarizing urban biodiversity values at a *strategic* level of urban assessments (*sensu* Gordon et al., 2009).

If systematically collected spatial data on richness and abundance of some taxa is available, it should preferably be used directly as input in spatial prioritization. If comprehensive data on richness and abundances of species is available, all other Biodiversity Quality attributes can be derived from that data, at least for animals for which biomass of individuals remains roughly constant within species (Feest et al., 2011). Vegetation biomass can be estimated from remote-sensing sources (Nichol and Lee, 2005). Only regional representativeness requires some kind of expert judgement, unless similarly-collected species data is also available from the surrounding regions.

Our results can be used to inform urban planning in the Helsinki Metropolitan Area. The highest biodiversity remains in the large forests at the urban fringe, but the method also identifies as important other types of urban green that contribute greatly to the urban biodiversity (Fig. 4). We also use the impact avoidance principle to identify the most suitable places for new urban developments (Kareksela et al., 2013). Infill-development and densification, which are both greatly emphasized in the current urban planning discussion in the Helsinki Metropolitan Area (e.g. The Helsinki City Plan 2050, 2016), are generally supported by our analysis (Fig. 6). Naturally, different land-use policies in the municipalities of the area, a high number of private stakeholders, and variable land-use planning goals, complicate the accounting of ecology in zoning and land use planning. The Biodiversity Quality potential maps (Fig. 3), together with the original scores made by the experts, may to some extent be used in mitigating trade-offs between different taxa in green area management, as the importance of the biotopes varies between different taxa. In the case of the Helsinki Metropolitan Area, for example, coastal meadows are extremely important for birds (especially specialist species), mammals and beetles, which should be accounted for in their management, whereas in natural sands, hymenoptera are more relevant.

Spatial prioritization analyses can integrate large data effectively, they can produce cost-efficient plans and can integrate many aspects of biodiversity and ecosystem services (Bekessy et al., 2012; Cimon-Morin and Poulin, 2018), which makes them useful for urban planning that respects the biodiversity-for-the-city paradigm (Pickett et al., 2016). However, to provide a truly holistic understanding of urban biodiversity values, residents' preferences and use of green spaces should be incorporated into the prioritizations (Pickett et al., 2016). For example, expert scoring could be done for sociotopes (i.e. urban areas with distinctive human use patterns) instead of biotopes (Lindholst et al., 2015). Another possibility is to determine taxon-specific responses to increasing recreational uses of different biotope types and include them as so-called condition layers in Zonation (Kujala et al., 2018b). Furthermore, prioritizations like the present one could be easily expanded by including e.g. layers describing the provision of, and demand for, ecosystem services (Cimon-Morin and Poulin, 2018), existing protected areas (Gordon et al., 2009), and road accessibility (Bekessy et al., 2012) to the analysis. In those types of analyses that include many types of social and ecological input data, the weighting system must be balanced between considerations, preferably by experts from many disciplines and stakeholder groups. Because cities do harbor also rare and endangered species (Gordon et al., 2009; Kowarik, 2011) that should not be completely neglected, individual target species and habitats can also be incorporated to analyses as new additional input layers (Moilanen et al., 2011). Penalties can be given for areas that are considered to be reduced in their potential for urban biodiversity, e.g. due to invasive alien species or other pressures. Technically, penalties can be implemented via condition layers, via negative connectivity interactions, or by including the harmful features as negatively-weighted input layers (Moilanen et al., 2011; Kujala et al., 2018b). All such additions are, of course, dependent on the availability of relevant spatial data.

This study demonstrates how the Biodiversity Quality framework can deepen our understanding about biodiversity in different biotopes or habitat types that often act as the basis of biodiversity plans in cities (Nilon et al., 2017). Basing the analysis systematically on different community attributes helps us to understand the differences between species assemblages found in different urban biotopes, which can consequently lead to high coverage of distinctive ecological communities in urban biodiversity conservation. For example, our case study suggests that botanical gardens have high value for urban biodiversity, equal to that of old-growth natural forests. We suggest that adding two attributes to the analyses, i.e. support for habitat specialists and regional representativeness, makes the framework even more relevant for urban biodiversity analyses. Although our method is restricted to the biophysical realm of urban areas and thus lies within the biodiversity-in-the-city paradigm (Pickett et al., 2016), it can nonetheless inform urban conservation and land-use planning about biodiversity values and their distribution within a given city. Importantly, our approach can be complemented with data about the human dimension of urban environments in a flexible manner, thereby enabling prioritizations that are compatible with the biodiversity-of and for-the-city paradigms (Pickett et al., 2016). During preliminary discussions, local urban planners, environmental authorities, and taxonomic experts have responded positively to the proposed approach of spatial prioritization of urban biodiversity, although this is not yet systematically verified.

5. Conclusion

The successful conservation of urban biodiversity depends on three things: a clear understanding of what should actually be conserved, planning methods that enable efficient conservation outcomes, and data that is adequately representative. Conservation planning tools, such as spatial prioritization, have much to offer for urban land-use and green infrastructure planning, as spatial prioritization can account for many aspects of biodiversity simultaneously. Hence, spatial prioritization can support urban land-use and green infrastructure planning, aimed at maintaining urban ecosystems and biodiversity.

Declaration of Competing Interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.ufug.2020.126586.

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